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Groundwater resources in the Jabal Al Hass region, northwest Syria: an assessment of past use and future potential

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Abstract In many cases, the development of groundwater resources to boost agricultural production in dry areas has led to a continuous decline in groundwater levels; this has called into question the sustainability of such exploitation. In developing countries, limited budgets and scarce hydrological data often do not allow groundwater resources to be assessed through groundwater modeling. A case study is presented of a low-cost water-balance approach to groundwater resource assessments in a 1,550 km^2 semi-arid region in northwestern Syria. The past development of irrigated agriculture and its effect on the groundwater system were studied by analysis of Landsat images and long-term groundwater level changes, respectively. All components of the groundwater balance were determined. Groundwater recharge was estimated using the chloride mass balance method. Over the past three decades, groundwater levels have declined, on average, 23 m, coinciding with a two-fold increase in the groundwater-irrigated area. Groundwater resources are currently depleted by a value that lies between 9.5×10^6 and 118×10^6 m^3 year^{-1}, which is larger than can be compensated for by a future decrease in natural discharge or changes in boundary conditions. However, groundwater resources are likely to be sufficient to supply domestic and livestock needs in the area.

Résumé Dans plusieurs cas, le développement des ressources en eau souterraine pour augmenter les productions agricoles dans les zones sèches a mené à un déclin continu des niveaux piézométriques ; ceci remet en question la durabilité de telles exploitations. Dans les pays en voie de développement, les budgets limités et le peu de données hydrologiques ne permettent pas souvent de dresser le bilan des ressources en eau souterraine à travers la modélisation hydrogéologique. Un cas d'étude est présenté, reprenant une approche par bilan hydrologique peu coûteuse pour évaluer les eaux souterraines dans une région semi-aride de 1550 km^2 du Nord-Ouest de la Syrie. L'historique du développement de l'agriculture irriguée et les effets sur le système hydrogéologique ont été analysés au moyen d'images Landsat et du changement sur le long terme des niveaux piézométriques. Toutes les composantes du bilan hydrologiques ont été déterminées. La recharge de l'eau souterraine a été estimée suivant la méthode du bilan massique des chlorures. Sur les trois dernières décennies, les niveaux piézométriques ont diminué - en moyenne de 23 mètres - ce qui coïncide avec une augmentation croissante de la surface irriguée par de l'eau souterraine. Les ressources en eau souterraine sont pour l'instant diminuées d'un taux situé entre 9.5×10^6 et 118×10^6 m^{3}an^{-1}, ce qui est largement supérieur à ce qui peut être compensé par une diminution de la recharge naturelle ou des changements des conditions aux limites. Toutefois, les ressources en eau souterraine sont suffisantes pour supporter l'alimentation domestique et les besoins des élevages dans la zone concernée.

Resumen En muchos casos, el desarrollo de recursos del agua subterránea, para incrementar la producción agrícola en las áreas secas, ha llevado a un descenso continuo en los niveles del agua subterránea; esto ha originado cuestionamientos acerca de la sostenibilidad de tal explotación. En los países en vías de desarrollo, a menudo, tanto los presupuestos limitados como los datos hidrológicos escasos, no permiten evaluar los recursos del agua subterránea a través del modelamiento de agua subterránea. Se presenta un estudio de caso, de un método económico para realizar un balance hídrico, destinado a evaluar el recurso de agua subterránea en una región semiárida de 1550 km^2, en el noroeste de Siria. El desarrollo pasado de agricultura irrigada y su efecto en el sistema del agua subterránea, fueron estudiados mediante el análisis de imágenes de Landsat y de los cambios de largo plazo en los niveles de agua subterránea, respectivamente. Todos los componentes del balance del agua subterránea fueron determinados. Se estimó la

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recarga de agua subterránea, mediante el uso del método de balance de masas de cloruros. Durante las últimas tres décadas, los niveles del agua subterránea han descendido, en el promedio 23 m, coincidiendo con un aumento, en dos órdenes de magnitud, en el área irrigada con agua subterránea. Los recursos de Agua subterránea son gastados actualmente en cantidades que oscilan entre 9.5×10^6 y 118×10^6 m^3/año, las cuales son mayores que lo que puede compensarse por una disminución futura en la descarga natural o por cambios en las condiciones de contorno del acuífero. Sin embargo, es probable que los recursos del agua subterránea sean suficientes para suministrar el consumo doméstico y del ganado que se necesitan en el área.

Keywords Groundwater recharge/water budget · Groundwater development · Agriculture · Semi-arid regions · Syria

Introduction

Management of groundwater resources requires knowledge of the reaction of a groundwater system to external stresses such as pumping. Such knowledge is especially critical in semi-arid regions, where natural recharge and discharge rates are often low, and groundwater pumping can rapidly dominate the behavior of the system. In many semi-arid regions, the development of groundwater resources has led to a continuous decline in groundwater levels, as documented by Postel (1999) and Shah et al. (2000). This raises the question to what extent groundwater exploitation in these regions is sustainable.

The relationship between declining water levels and the sustainability of groundwater exploitation is not clear-cut. Falling water levels could be due to the overexploitation of resources; however, they could also be the result of a transient move towards a new equilibrium in the groundwater system following an increase in pumping (Custodio 2002). The sustainability of groundwater exploitation can be evaluated using water-balance or groundwater-modeling studies. The use of water-balance approaches to determine sustainable pumping rates is embedded in groundwater legislation in many countries (Kalf and Woolley 2005). It is often assumed that pumping rates are sustainable if they are less than the recharge (Kalf and Woolley 2005). Several authors have pointed out that this approach oversimplifies the groundwater system (Bredehoeft et al. 1982; Bredehoeft 2002), because it ignores the effects of pumping on recharge and discharge rates and boundary conditions. In addition, questions have also been raised as to whether sustainable pumping rates (defined as pumping rates that allow a long-term balance to be achieved between recharge and discharge), are an appropriate measure for sustainable groundwater development. As several authors argue, sustainable groundwater development does not simply involve stabilizing water levels alone: it also involves factors related to water quality, ecology and socio-economic considerations (Sophocleous 1997; FAO 2003; Alley and Leake 2004; Maimone 2004).

As argued by Bredehoeft (2002), groundwater modeling is generally the preferred approach when studying a groundwater system's response to pumping. This is because it can include, explicitly, the spatio-temporal nature of the response of the system to imposed stresses, as well as the changes in boundary conditions and recharge and discharge rates that result from this response. However, groundwater modeling studies require far more resources, in terms of both budgets and hydrological data, than water-balance studies do. In much of the developing world, research budgets are limited. In addition, hydrological data are often scarce, which impairs attempts to calibrate groundwater models. Therefore, water-balance approaches are often a more viable alternative for groundwater-assessment studies. As discussed above, the water-balance approach has certain limitations when used to determine sustainable pumping rates. This makes it necessary to evaluate (1) the use of this method in groundwater-assessment studies and (2) the conclusions that can be drawn from such studies regarding sustainable groundwater management.

Recharge is one of the key components of the groundwater balance. Most authors recommend the application of multiple techniques to estimate recharge (Alley et al. 2002; De Vries and Simmers 2002; Scanlon et al. 2002). However, relatively few methods can be applied to estimate the regional, long-term average values that are required for water-balance studies. Unsaturated zone techniques (lysimeters, chloride profiles, Darcy based methods) provide point estimates or estimates of recharge over relatively small areas, whereas water-table fluctuation methods provide estimates that range from event timescales to the length of the hydrographic record. Regional long-term integrated estimates can be obtained by methods based on the use of chemical or isotopic tracers in the saturated zone (Scanlon et al. 2002). Methods based on the analysis of stable isotopes (^{18}O, ^{2}H) can provide information on recharge processes and the location of recharge areas, while analysis of radioactive isotopes (^{3}H, ^{3}He, ^{14}C) can be used to estimate recharge by dating groundwater (De Vries and Simmers 2002; Scanlon et al. 2002). However, in many developing countries, standard isotope sampling and analyses are lacking and scientists often do not have access to laboratories that can perform isotope analysis. The chloride mass balance (CMB) method, which is based on the use of chloride as a tracer of recharge, has been applied to provide regional estimates of long-term average groundwater recharge in a large number of studies in semi-arid regions; examples include Dettinger (1989), Sami and Hughes (1995), Bazuhair and Wood (1996) and De Vries et al. (2000). Use of the CMB method can provide estimates of very low recharge rates; the method's accuracy actually increases as recharge rates decrease (Scanlon et al. 2002). Therefore, the CMB method was judged to be the most suitable for integration into a low-cost groundwater assessment in a semi-arid environment.

Groundwater pumping in the semi-arid regions of Syria has increased significantly over the last two decades, due

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to the development of groundwater-based irrigated agriculture. For example, the irrigated area in regions where precipitation does not exceed 350 mm year^{-1} doubled between 1985 and 2004 (Ministry of Agriculture and Agrarian Reform; MAAR 2006). Though an increase was observed in the extent of areas irrigated with surface water such as the Euphrates irrigation projects, the percentage of the total irrigated area served by groundwater rose from 49% in 1985 to 60% in 2004 (MAAR 2006). In his investigations of the groundwater resources of the Aleppo basin in northern Syria, Luijendijk (2003) found that the upper aquifer system was over-exploited. Syria's high population growth rate, currently estimated to be 2.9% per year (CBS 2003), is expected to increase the pressure that irrigated agriculture and domestic needs exert on groundwater resources in the future. This raises the need for information which can be used to determine whether the capacity of the groundwater system can sustain both irrigated agriculture and domestic water use in the long term.

The objective of this study was to explore the feasibility and limitations of a low-cost, water-balance-based approach for groundwater resource assessments in a semi-arid environment. The study focused on a method that can be applied in catchments in which hydrological information is scarce, and in which more data-intensive or sophisticated hydrological studies such as long-term groundwater monitoring programs or groundwater modeling studies, are not feasible. The study was conducted in the semi-arid Jabal Al Hass region in northwestern Syria.

Materials and methods

Physiography and climate

The study area covers an area of 1,550 km^2, located in central northwestern Syria, at a distance of 100–150 km from the Mediterranean Sea. The area is located in the transitional zone between the relatively wet coastal zone and the dry steppe and desert areas of central and southern Syria and is typical of the marginal semi-arid agricultural areas of Syria. The regional setting of the area and the main topographic features are shown in Fig. 1.

The study area is dominated by the Jabal Al Hass basalt plateau, which is located at an elevation of 350–600 m above sea level (masl). This plateau is surrounded by valleys at elevations of 250–300 masl. The Qwayq River Valley is located west of the plateau. A network of irrigation canals is used to distribute river water to an area of approximately 200 km^2 in the valley. Surface water in the study area drains towards the south to four "sabkhas" (salt flats).

The climate of the study area is semi-arid, and has a short, relatively wet winter and a long dry summer. The average monthly temperature ranges from 6°C in January to 28°C in August (International Center for Agricultural Research in the Dry Areas, ICARDA, unpublished data,

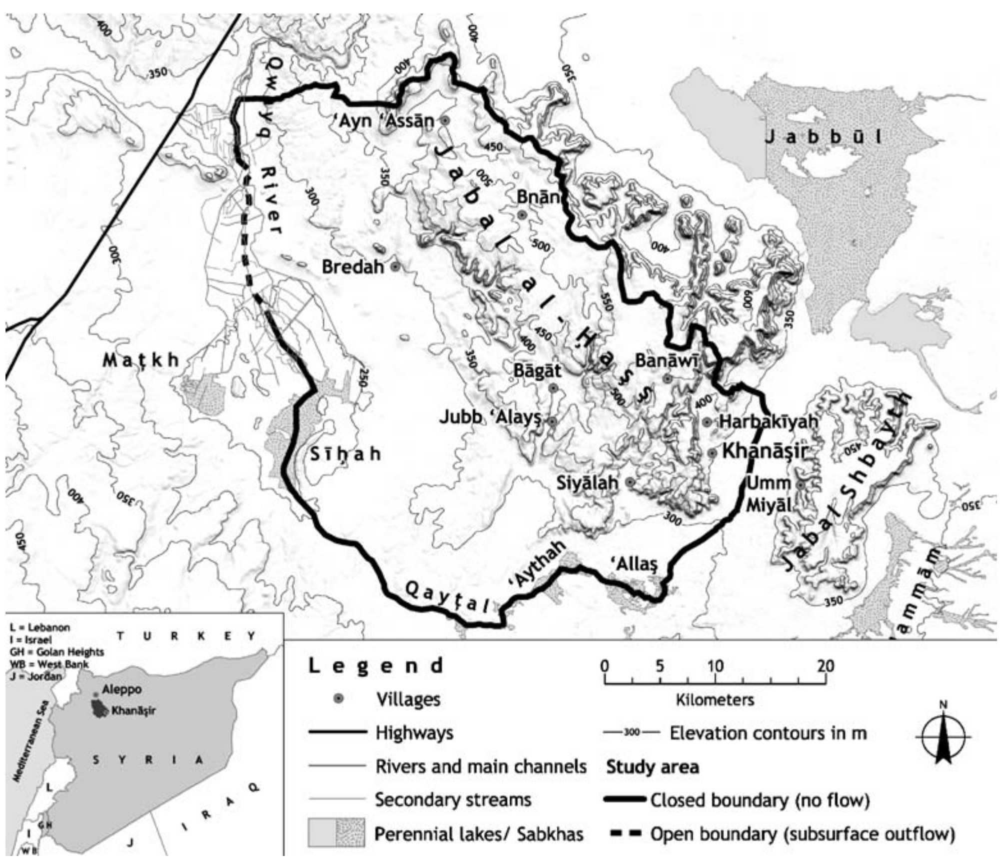

Fig. 1 Location and main topographic features of the study area

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2006). Rainfall occurs almost exclusively between November and April. Long-term average precipitation varies between 350 mm in the west of the study area to 210 mm in the southeast of the area (ICARDA, unpublished data, 2006). Reference evapotranspiration at a climate station in the center of the study area (Bredah) (1993–1999), calculated using the FAO Penman-Monteith equation (Allen et al. 1998), was 1,700 mm year^{-1}.

Hydrogeological setting

The geology and hydrogeology of the area, as summarized below, were described by Technoexport (1966) and Gruzgiprovodkhoz (1982), respectively. See Fig. 2 for an overview of the geology of the study area and Fig. 3 for a geological cross-section. Two aquifers can be identified in the study area. The deep aquifer is confined and its top is located at a depth of 250 m or more. This aquifer consists of an approximately 500-m-thick sequence of Upper Cretaceous limestones. The deep aquifer is overlain by a 90–215-m-thick formation of Maastrichtian (Upper Cretaceous) marls, clays and silty limestones. This formation acts as a confining layer.

The upper, unconfined, aquifer mainly consists of chalky Paleocene and Eocene limestones. The thickness of this formation varies from 20 m in the eastern section of the study area to 200 m in the Jabal Al Hass area and the Qwayq Valley. West of the basalt plateau, this formation is overlain by an 80-m thick layer of fissured and karstified coarse limestones, interbedded with layers of basalt, which is classified as the Helvetian Formation (middle Miocene). The Jabal Al Hass plateau is covered

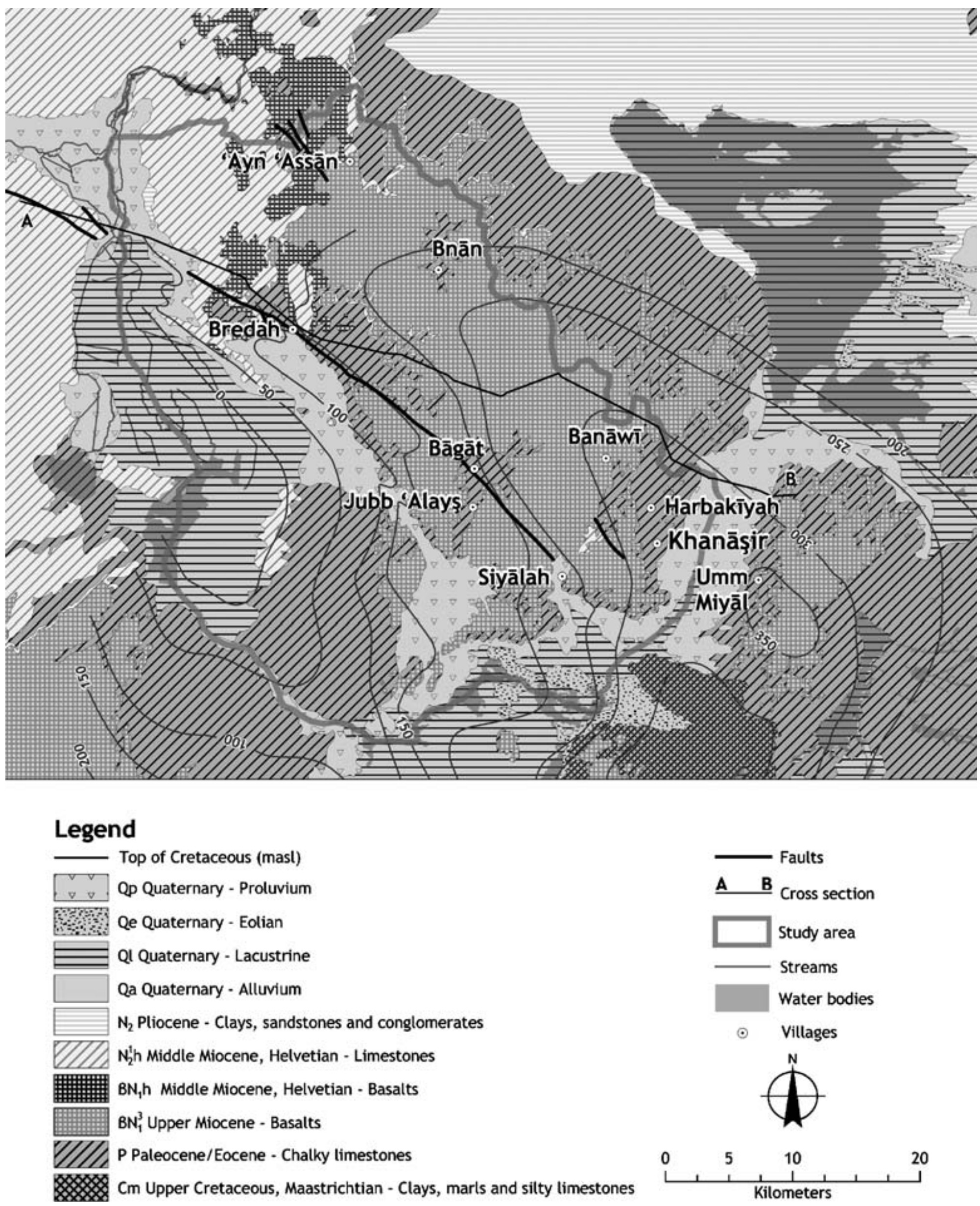

Fig. 2 Geological map and contour lines of the top of the Cretaceous formation. Modified from Technoexport (1966)

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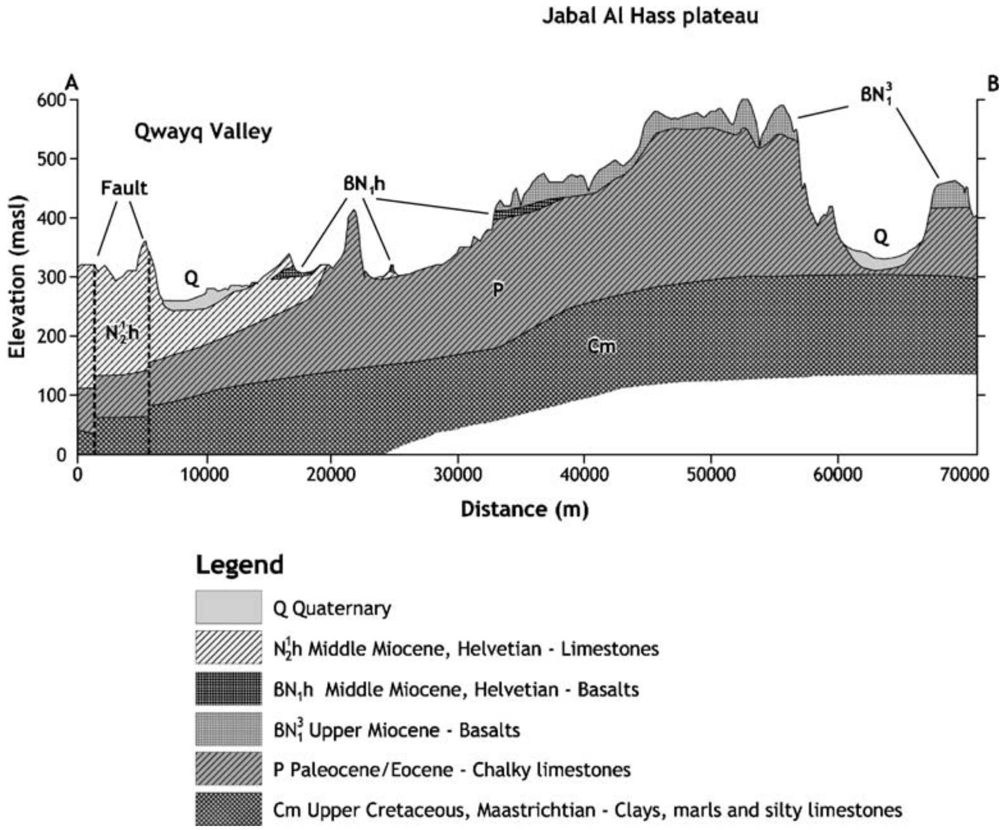
Fig. 3 Geological cross section *A–B*. See Fig. 2 for the location of this section



by a strongly jointed upper Miocene basalt sheet, with an average thickness of 10 m. A 20–50-m-thick sequence of Pliocene and Quaternary sedimentary deposits is found in the valleys, consisting of alluvial sands and conglomerates; and lacustrine clays, loam and gypsum.

Field survey and data collection

Data on groundwater levels, groundwater quality, well capacities and water use were obtained from a field survey conducted between June and September 2004. The wells surveyed were mainly boreholes, although a small number of large (1-m diameter) wells, generally known as ‘Arabic wells’, were also studied. The well survey was restricted to wells that tap the upper aquifer. The depth of the wells ranged from 20 to 225 m, and averaged 70 m. Well locations were recorded using a global positioning system (GPS), with a spatial accuracy of 5–10 m. The surface elevations of the wells were obtained from geo-referenced topographic maps with a scale of 1:50,000 and a contour interval of 5–10 m. Groundwater levels were measured with an electro-probe in 76 unused or abandoned wells. Groundwater level contour maps were constructed using ordinary kriging (Krige 1951). In 139 cases, measurements were made of the electrical conductivity (EC) and temperature of samples of groundwater pumped from recently used wells. Some general observations were also made, including the type of well and the type of pump used, the diameter of the pipe and the land use in the area around the well, and farmers were asked about past and future groundwater use.

In total, 44 groundwater samples from freshly pumped, unlined wells were collected in PVC bottles so that their major chemical constituents could be determined in the laboratory. The alkalinity of the samples was determined in the field using the Gran titration method (Appelo and Postma 1996). The samples were acidified to a pH of less than 2 through the addition of 0.7 mL 1N HNO_3 for preservation. Major ions were analyzed using standard procedures documented by Ryan et al. (2001). Na^+ and K^+ were determined by flame photometry, Ca^{2+}, Mg^{2+}, NO_3^- and Cl^- were analyzed using titration and SO_4^{2-} was determined by barium sulfate precipitation. Seven samples were rejected, because the electro-neutrality balance (Appelo and Postma 1996) exceeded 5%. Water types were assigned according to the dominant cations and anions, where the dominant anion/cation was taken to be the ion which accounted for the greatest percentage of the total ions, as expressed in meq L^{-1}. In cases where the differences between ions were lower than 5%, more than one cation or anion was listed as the dominant ion.

The Cl^- concentration of precipitation was measured in the 2001–2002 and the 2002–2003 rainy seasons (winter) in two locations in the eastern part of the study area (Abou Zakhem 2008). In addition, two monthly precipitation samples taken in the 1989/1990 winter season near the city of Aleppo, 30 km north of the study area, were used (Kattan 1996). The location at which groundwater and

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precipitation samples were collected is presented in Fig. 4. In both cases, precipitation was collected in a large open container, with a 40-cm diameter, which was emptied in a closed polyethylene storage container after each rainfall event. Each month, the precipitation was analyzed for major chemical constituents. All ions were determined using ion-chromatography, with the exception of HCO_3^-, which was determined by titration. The methods of sampling and analysis were documented by Kattan (1996) and Abou Zakhem (2008). Samples were rejected in cases where the electro-neutrality balance exceeded 10%. The Cl^- concentration of the bulk precipitation samples was the result of both wet and dry deposition during the rainy season.

Assessment of groundwater level change

Information on past groundwater levels was obtained from two previous hydrogeological studies: Technoexport (1966) and Gruzgiprovodkhoz (1982). These authors reported the groundwater levels of 13 observation wells in meters below ground level (mbgl), with the locations of these wells provided on a 1:250,000 scale map. A map of the current groundwater level of the study area was prepared from the well water levels observed during a field survey (see section on field survey). To address the uncertainty caused by locating the historic observation wells, the minimum and maximum groundwater levels within a 1-km radius of the location of each historic well were identified from the current groundwater level map.

Analysis of the development of irrigated agriculture

The development of irrigated agriculture was reconstructed through the analysis of the normalized difference vegetation index (NDVI) of seven Landsat images (Rouse et al. 1973). Four images were used to estimate irrigated areas in the summer season: September 1975, July/August 1985, September 1999 and September 2000, and three images were used to estimate the irrigated area in the winter season: April 1987, April 1995, March 2003. The pixel size of the satellite images from September 1975 and July/August 1985 was 67 m, the pixel size of the image from April 1987 was 57 m, and the pixel size of the remainder of the images was 30 m.

The NDVI thresholds, which were used to differentiate between irrigated and non-irrigated areas, were deter-

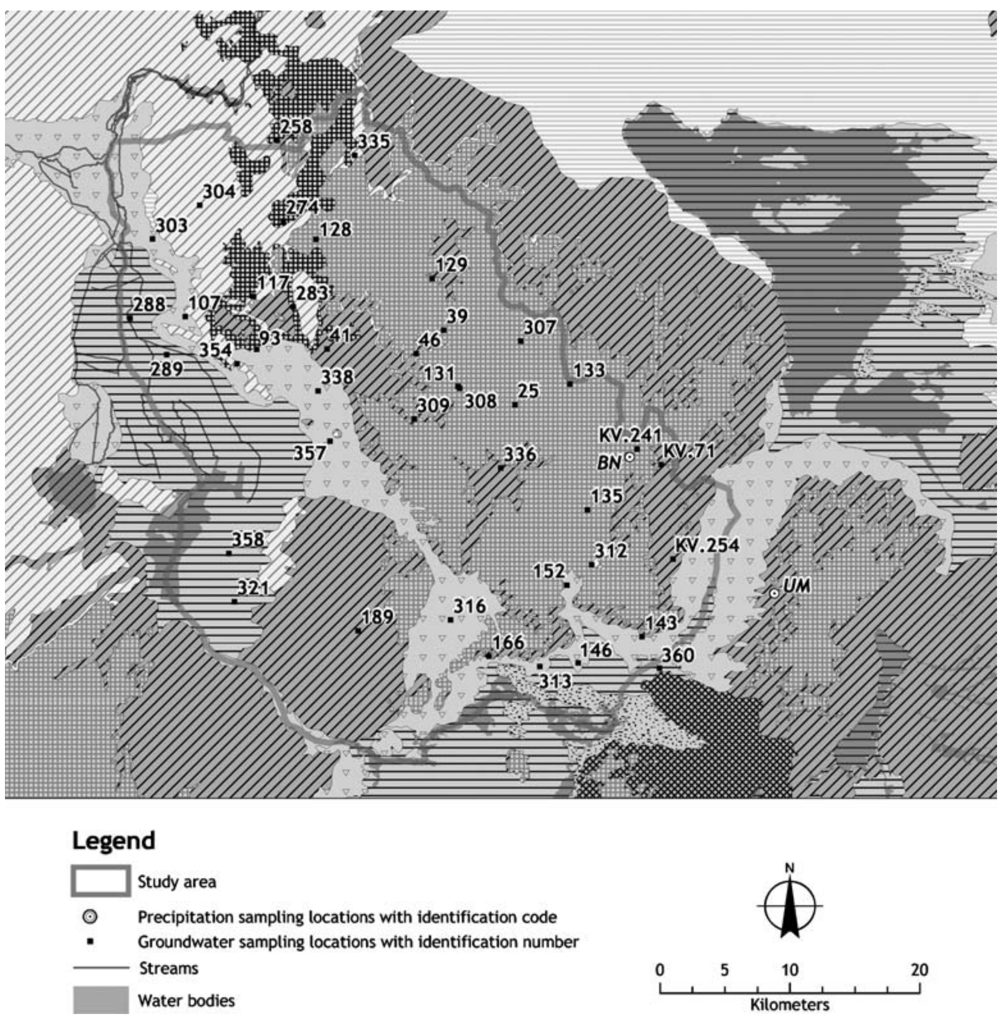

Fig. 4 Locations at which groundwater and precipitation samples were collected in the study area. See Fig. 2 for legend of geology

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mined considering the location of irrigation wells and fields mapped during the field survey, the surface drainage network, and expert knowledge of the area. The often non-uniform application of irrigation in the study area and the fact that the northwestern part of the study area receives more rainfall than the southeastern part, resulting in higher NDVI values for non-irrigated areas in the northwest, significantly complicated the determination of the threshold value. To address these uncertainties, three NDVI thresholds were chosen. The minimum NDVI threshold included all irrigated areas but also a significant number of pixels that had received water from "wadis" (ephemeral streams) or rocky areas. The maximum NDVI threshold excluded all areas that had received surface runoff, but thereby also excluded those fields that by not having been uniformly irrigated, were indistinguishable either from flooded areas or from rain-fed areas in parts of the study area with adequate rainfall. The best estimate resulted from a compromise between excluding pixels that belonged to an irrigated field and including pixels that had received surface runoff. The minimum estimate of the irrigated area was computed using the maximum estimate of the NDVI threshold and vice versa.

Groundwater balance

Changes in groundwater storage due to groundwater abstraction were estimated using a groundwater balance of the upper aquifer. The following water balance equation was constructed for the study area:

$$\frac{dV}{dT} = R - (NP + ET + Q) \tag{1}$$

where dV/dT is the change in storage (m^3 year^{-1}), R is groundwater recharge (m^3 year^{-1}), NP is net groundwater pumping, which equals groundwater pumping minus all return flows (m^3 year^{-1}), ET is evaporation and transpiration of groundwater (m^3 year^{-1}) and Q is cross-boundary groundwater outflow (m^3 year^{-1}).

The water balance was calculated for an average current year, using recent studies and data collected during the years 2000–2003. The rainfall during the years used for the assessment of irrigation water use in the winter season was near the long-term average, whereas irrigation water use in the summer was assumed to be independent of rainfall, because almost no rain falls during the summer season. The uncertainty of parameter values was addressed by estimating relative uncertainties and minimum and maximum parameter values. These estimates were based on expert judgment.

The location of the boundaries of the area within which the groundwater balance was calculated is presented in Fig. 1. The northern and eastern boundaries of the area consist of the Jabal Al Hass groundwater divide and the bottom of the Khanasir Valley. The western and southern boundaries coincide with the Qwayq River, the bottom of the Sihah, Qaytal, Aythah and Allas sabkhas, and the valleys connecting these sabkhas. The field survey revealed that the lowest hydraulic heads are found at the study area boundaries. No hydraulic gradient from the boundaries into the study area could be observed, with the exception of the boundary located in the Qwayq River Valley. Therefore, all boundaries except the Qwayq River boundary were considered as no-flow boundaries.

Groundwater balance components

Recharge

The area in which groundwater recharge (R) occurs was delineated using information on soils and geology, as well as information on groundwater levels and groundwater composition. The long-term average groundwater recharge in these areas was estimated using the chloride mass balance method.

The chloride mass balance method uses the chloride ion (Cl^-) as an environmental tracer. Provided that Cl^- containing sediments such as halite are absent, all the Cl^- in a hydrological system is derived from atmospheric input, e.g., wet and dry deposition. The Cl^- ion can be considered to be conservative, as it does not react with constituents in the soil or in the aquifer (Hem 1985). Chloride is also not taken up by vegetation or removed from the system by evaporation. Because of this, it can be used to solve mass balance equations that relate precipitation to groundwater recharge. The equation used in this study was a modified version of that given by Edmunds et al. (1988), i.e.,

$$R^* = \frac{C_b P + D}{C_{gw}} \tag{2}$$

where R^* is recharge (mm year^{-1}), C_b is the bulk (wet and dry) Cl^- deposition during the winter, expressed in mg Cl^- per liter of precipitation, C_{gw} is the Cl^- concentration in groundwater (mg L^{-1}), P is precipitation (mm year^{-1}), and D is the dry deposition of Cl^- during the summer (mg m^{-2} year^{-1}). Note that R^* denotes recharge in one spatial dimension, as opposed to the value of recharge (R) in Eq. 1, which is given in three spatial dimensions.

The CMB method provides spatially integrated recharge estimates over an area upgradient from the measurement point (Scanlon et al. 2002). The method gives an estimate of the average long-term recharge, because temporal differences in Cl^- concentration in the groundwater recharge flux that result from temporal changes in the magnitude of groundwater recharge are obscured by multi-modal flow, diffusion, dispersion and mixing processes in both the unsaturated and saturated zone. Groundwater samples, which were analyzed for Cl^- and other chemical constituents, were taken from freshly pumped, unlined wells. Thus, these samples represent a mixture of the groundwater in the penetrated portion of the aquifer.

It should be noted, however, that the CMB method is only valid if the following conditions are met:

1. The Cl^- concentration of groundwater should be controlled by the composition of infiltrating recharge.

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2. All Cl^- deposited on the surface should be leached by precipitation.
3. There should be no significant net surface runoff or run-on of water in the selected area.
4. There should be no significant Cl^- deposition from human and livestock urine and excreta.

The validity of the first condition was verified using baseline information on hydraulic heads and groundwater composition, as well as information obtained from the relevant literature on the geology, geomorphology and soils of the area. With regard to the second condition, the fact that NaCl is highly soluble makes it very unlikely that any buildup of Cl^- could occur in the surface layer; all Cl^- was therefore assumed to be leached by precipitation.

Runoff observations indicated that wadi runoff is only generated by precipitation events of 20 mm day^{-1} or more, which occur 1.3 times per year on average. The predominant barley crop on the well-drained, clay loam soils would be represented by a runoff curve number value of 85 (USDA-NRCS 1986), which would generate 2.2 mm surface runoff at 20 mm rain. Using this curve number for the 25-year daily precipitation record of the Bredah station (ICARDA, unpublished data, 2006) gives an average annual surface runoff of 7.2 mm, which is 2.6% of the average annual precipitation of 275 mm. Thus, the influence of runoff on the chloride mass balance calculation was assumed sufficiently small to be neglected.

Although the population of the area has increased steadily with time, with 213 inhabitants per km^2 in 2004 and 390 small ruminants per km^2, the population density is still low. Therefore, the influence of human and livestock urine and excreta on the chloride mass balance was assumed to be negligible. Long-term average rainfall (P) in the study area was calculated using the 1-km resolution climatic grid of Syria developed by De Pauw et al. (2001). The uncertainty of the long-term average precipitation and the area of recharge were assumed to be relatively low, 10%.

The uncertainties associated with the bulk Cl^- deposition during the winter months are related to the sampling method, the laboratory analysis, and the spatial and temporal variations of deposition. The capture of wet deposition is influenced by aerodynamic effects, which may cause underestimation of the capture of fine raindrops that tend to have higher than average solute concentrations (Thimonier 1997). An additional source of uncertainty is the fact that the dry deposition captured by the bulk precipitation samplers may not be representative of the dry deposition to natural surfaces, due to the strong influence of the vegetation on dry deposition (Erisman et al. 1994; Wesely and Hicks 2000). Furthermore, the high spatial and temporal variability of wet and especially dry deposition (e.g., Harrison et al. 2001) may not have been completely captured by the 11 monthly samples from 3 stations used for this study. Average annual bulk chloride concentration observed at stations with similar annual precipitation and distance from the coast as the study area ranged from 2 to 6 mg L^{-1} during the 1989/1990 rainfall season for five stations in Syria (Kattan 1996). Similarly, Hingston and Gailitis (1976) found average chloride concentrations between 1.0 and 4.5 mg L^{-1} for 9 stations during a 1-year monitoring period in Western Australia, whereas Blackburn and McLeod (1983) reported average concentrations of 1.5–3.9 mg L^{-1} for a 3-year sampling period at three stations in southeastern Australia. Considering the above, it was assumed that a range of 2–6 mg L^{-1} would cover the uncertainty of the bulk chloride deposition during the winter season.

As measurements of dry deposition during the summer months were not available, dry deposition was estimated using a physical model (Appelo and Postma 1996). According to Eriksson (1959), dry deposition of aerosols can be calculated using the following equation:

$$D = 3.15 \cdot 10^5 v_d C \tag{3}$$

where D is the dry deposition (μg m^{-2} year^{-1}), v_d is the dry deposition velocity (cm s^{-1}) and C is the atmospheric concentration of the aerosol (μg m^{-3}). A representative value of the atmospheric concentration of Cl^-, 2.2 μg m^{-3}, was derived from Bardouki et al. (2003). Dry deposition at distances of more than 100 km from the coast is caused mainly by particles of 0.1–0.8 μm in diameter (Möller 1990). Variations of approximately an order of magnitude in field measurements of dry deposition velocities have been reported in the literature (Gallagher et al. 1997). In a review of studies that compared measured and modeled dry deposition velocities Wesely and Hicks (2000) found an uncertainty of 30% in computed dry deposition velocities, while Garland (2001) reports discrepancies of more than an order of magnitude between theoretical predictions and measured dry deposition velocities of particles in the range of 0.1–1.0 μm, especially in forested areas. Based on values reported in the literature (Mészáros 1981; Möller 1990), 1 cm s^{-1} was selected as the best estimate of the dry deposition velocity of chloride, and the lowest and highest values of the dry-deposition velocity found in the literature, i.e., 0.2 and 2 cm s^{-1}, were used to calculate minimum and maximum estimates of dry deposition. Because these values have a large range, it was assumed that their use would account for all the uncertainty associated with the estimate of dry deposition. The relative uncertainty of the chloride concentration in groundwater samples was estimated to be 25%. This uncertainty stems mainly from analytical errors, and potential errors due to an incomplete spatial coverage of the sampling stations.

Net pumping

Groundwater pumping was calculated for all different types of water use. Groundwater pumping for irrigated agriculture is widespread in the area, and nearly all farmers have drilled one or more wells on their property during the last two decades. In the winter cropping season, which lasts from November to May, rain-fed crops such as

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barley and wheat, are grown. These crops usually receive supplemental irrigation if sources of groundwater or surface water are available, and this is especially true in the case of wheat. During the summer, farmers grow cotton, maize and several types of vegetables. These crops rely completely on irrigation. In the center of the Qwayq Valley, surface water originating from the Qwayq River is used for irrigation. During the summer months, however, this river dries up, forcing the farmers in the southern part of the basin to irrigate with groundwater.

As described elsewhere in this report, the area irrigated during the summer and winter season was mapped by analyzing satellite images. The area in which surface water rather than groundwater was used for irrigation was delineated with the help of a field survey. The relative aerial coverage of the different irrigated crops was derived from the Ministry of Agriculture and Agrarian Reform (MAAR 2006). Estimates of the irrigation applications for the different crops use were derived from various sources (Martin 1999; Hoogeveen et al. 1999; Aw-Hassan et al. 2003; Schweers et al. 2004). The amount of water farmers apply to crops is highly variable, due to the widely varying irrigation practices. The relative uncertainty of the crop water use was therefore estimated to equal 25%. Field observations indicated that an estimated 70% of the total volume of groundwater used was pumped from the upper aquifer and the remainder was pumped from the deep aquifer.

Information on domestic water use per capita was obtained by interviews in selected villages in the study area. Population data and data on the number of livestock were derived from data of the United Nations Development Programme (UNDP) Jabal Al Hass project (Omran and Breek 2000). The data were adjusted for population growth using the average annual rural population growth rate for the rural areas of Aleppo province of 2.9% (CBS 2003). The population and number of livestock of the study area were estimated by multiplying the data of Omran and Breek (2000) with the ratio between the study area (1,550 km^2) and the UNDP project area (1,300 km^2). Information on typical water use of livestock in a semi-arid climate was obtained from Heck (1995). The relative uncertainty of population and livestock data was estimated to be 5 and 10%, respectively.

The difference between the total amount of water pumped and the net pumping is caused by return flow of excess irrigation water. Previous research (Hoogeveen et al. 1999; Martin 1999; Aw-Hassan et al. 2003; Schweers et al. 2004) has shown that farmers in the area tend to irrigate approximately 20% more than the crop water requirements. However, only part of this water contributes to irrigation return flow, with evaporation likely to be the major source of inefficiency. The value of irrigation return flow was assumed to be relatively low, because the majority of irrigated fields were located in areas with thick clayey soils, low permeability bedrock (Paleocene/Eocene chalky limestones) and a water table that was located deep below the surface. The irrigation return flow was estimated to be equal to 5% of the irrigation water applied, with minimum and maximum estimates of 0 and 10%, respectively. Irrigation return flow was subtracted from the computed total amount of water pumped to compute the net pumping rate (NP). The computed irrigation return flow also includes irrigation return flow from fields that were irrigated with water from the Qwayq River or the associated irrigation canals.

Evaporation and transpiration

Evaporation and transpiration of groundwater takes place in areas where the water table is close to the surface, e.g., in and around the sabkhas. The current magnitude of this water-balance component is unknown. Before pumping began, all groundwater recharge in the system was balanced by evapotranspiration in and near the sabkhas and groundwater outflow into the Qwayq River. Currently, however, evapotranspiration is significantly lower and outflow into the Qwayq River has ceased, because pumping has lowered the hydraulic gradient and the groundwater flow towards the Qwayq River and the sabkhas. Although the present value of evapotranspiration is not known, this value must lie somewhere between zero and the original, pre-pumping, value of the total natural discharge (evapotranspiration plus outflow into the Qwayq River), which was equal to the recharge. Thus, the minimum estimate of evapotranspiration was assumed to be zero, whereas the maximum estimate was assumed to be equal to the groundwater recharge (R). The best estimate was somewhat arbitrarily chosen to lay halfway between the minimum and maximum estimate.

Cross boundary flow

The groundwater flow across the boundary in the Qwayq River Valley (Q) was estimated using Darcy's groundwater flow formula

$$Q = TL\frac{dh}{dx} \tag{4}$$

where Q is the volumetric water flow (m^3 d^{-1}), T is transmissivity (m^2 d^{-1}), L is the length of the boundary across which outflow occurs (m), h is the hydraulic head (m) and x is a measure of distance (m). The hydraulic gradient perpendicular to the boundary (0.0015) and the length of the boundary across which outflow occurs (15 km) were obtained from groundwater levels measured in the course of the field survey. Information on the thickness (80 m) of the Helvetian limestone formation and 11 observations of the hydraulic conductivity of this formation were obtained from Gruzgiprovodkhoz (1982). Considering the lognormal behavior of the hydraulic conductivity data, the median value of 0.24 m d^{-1} was used for the best estimate and the uncertainty of this component was accounted for by using the reported minimum (0.02 m d^{-1}) and maximum (2.24 m d^{-1}) values of hydraulic conductivity.

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Results

Groundwater level change

In most of the villages visited, the old, manually operated Arabic wells, which were the main source of groundwater prior to the introduction of the diesel pump, were found to be dry at the time of the field survey. Traditionally, “qanats” were also used to provide irrigation water in the area. These ancient subterranean aqueducts tap the groundwater stored in hills or in alluvial valleys, and consist of sloping underground channels constructed to provide water to agricultural fields and settlements (Lightfoot 1996). Pioneer farmers discovered and restored a qanat near the town of Khanasir in 1938; however, by 1977 the groundwater level had dropped below the base of the qanat and it had dried up. The qanat near the village of Jubb Alays dried up in the 1960s, a few years after the establishment of groundwater wells in the area, according to the villagers.

The location of the observation wells and qanats surveyed from July to September 2004 and the resulting groundwater-level contours are presented in Fig. 5. The groundwater levels roughly followed the topography of the study area. The depth of the water table ranged from 1 to 110 m below the ground surface. In the study area, groundwater flowed from the Jabal Al Hass basalt plateau towards surrounding valleys and depressions. A significant cone of depression of groundwater levels occurs in the Qwayq Valley, which extends west across the boundaries of the study area. In that area, groundwater levels are located approximately 70–80 m below the surface. By contrast, the groundwater level in the northern part of the Qwayq Valley and near the Sihah depression is located only a few meters below the surface.

Despite the high degree of uncertainty that is associated with the comparison of past and present groundwater levels of the 13 historic observation wells, as presented in Table 1, it is clear that in most wells the groundwater level

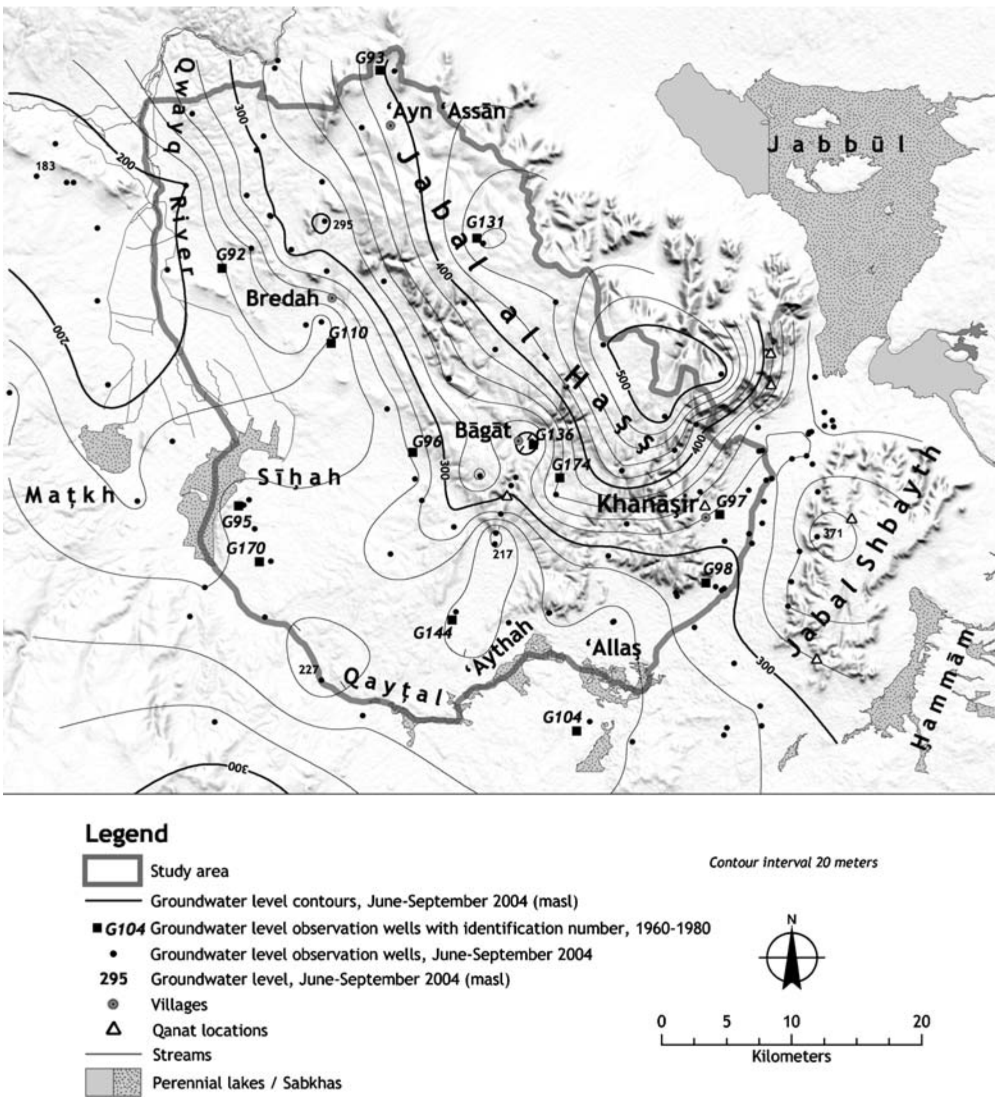

Fig. 5 Groundwater level of the upper aquifer (July–September 2004), the location of groundwater-level observation wells, and the location of qanats in the study area. The location of the groundwater level observation wells of 1960–1980 was obtained from Technoexport (1966) and Grughiprovodkhoz (1982)

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Table 1 Groundwater levels in observation wells during the period 1960 to 1980 and in 2004

Well no. (see Fig. 5)	Groundwater level, 1960–1980		Groundwater level, 2004		Groundwater level difference	
	Water level (mbgl)[a]	Source	Min. (mbgl)	Max. (mbgl)	Min. (m)	Max. (m)
G92	30	G	45	55	−15	−25
G93	8.5	T	30	50	−21.5	−41.5
G95	5	G	2	8	−3	3
G96	23.5	T	30	50	−6.5	−26.5
G97	19	G	17	35	2	−16
G98	15	T	20	30	−5	−15
G104	13	G	32	40	−19	−27
G110	12	G	45	65	−33	−53
G131	15	T	20	40	−5	−25
G136	12	T	50	70	−38	−58
G144	16	T	40	50	−24	−34
G170	6	G	9	12	−3	−6
G174	5	G	40	60	−35	−55
Average					−16	−29

[a] Meters below ground level (mbgl)

G according to Gruzgiprovodkhoz (1982); *T* according to Technoexport (1966)

has declined significantly over the course of the last three decades. In the observation wells used in this study, this decline averaged 23 m. The greatest decline (38–58 m) was recorded in an intensively irrigated valley in the Jabal Al Hass region (well G136, see Fig. 5).

Hydrochemistry

As presented in Fig. 6, the groundwater in the center of the valleys is highly saline: the ECs varied from 10 to 20 dS m^{-1}, while in the remainder of the study area the EC of groundwater ranged from 0.5 to 3 dS m^{-1}.

The chemical composition of the precipitation and groundwater samples are presented in Tables 2 and 3, respectively. The water type of the samples located in regions where the salinity is relatively high (EC in excess of 6 dS m^{-1}) such as the outcrop of Quaternary lacustrine sediments and adjacent areas (well 354, 357), is Na–Cl. The dominant water type in the remainder of the study area is Na–SO_4. The exception to this rule is formed by samples taken from wells in the Miocene Helvetian outcrop, in which the dominant cations are Ca^{2+} and Mg^{2+} and the dominant anions are SO_4^{2-} and HCO_3^-.

Development of irrigated agriculture

The results of the satellite image analysis, presented in Table 4 and Figs. 7 and 8, indicated a large increase in irrigated agriculture in the study area over the course of the last three decades. Expressed as a percentage of the aerial extent of the study area, the groundwater-irrigated area in the winter season increased from 1.9% in 1987 to 5.3% in 2003 (area A, Fig. 7). Because there is generally sufficient surface water available for irrigation in the Qwayq River Valley (areas B and C in Fig. 7) in the winter season, the irrigated fields in this area were not included in the estimates of the groundwater-irrigated area.

The total irrigated area in the summer season, including area B and C in Fig. 8, where both groundwater and surface water were used, changed from 2.0% in 1975 and 1.7% in 1985 to 2.9% in the year 2000. During the field survey it was found that in recent years during the summer season only the northern part of the valley (area B in Fig. 8) received sufficient water from the Qwayq River for irrigation, while in the southern part of the valley (area C in Fig. 8) all fields were irrigated with groundwater. To what extent this situation has changed since 1975 is not known. The minimum, best and maximum estimates of the groundwater-irrigated area in the southern part of the Qwayq Valley in September 2000 equaled 6.68, 14.17 and 25.82 km^2, respectively.

Groundwater balance components

Recharge

Delineation of recharge areas Favorable conditions for groundwater recharge were assumed to be limited to the outcrops of the Miocene basalt formation, also called the Jabal Al Hass basalt plateau, and the outcrop of the Helvetian limestone formation. The reasons for this assumption are outlined below. In both areas, soils are shallow and stony. In addition, in the Jabal Al Hass plateau the bedrock consists of a strongly jointed and weathered basalt layer. As several authors have pointed out, the recharge rates associated with basalt plateaus that have high levels of surface fracturing and weathering tend to be high (Athavale et al. 1983; De Vries and Simmers 2002). In the Helvetian limestone outcrop region, the bedrock consists of permeable, karstic and fissured limestone. In either case, a combination of shallow soil and highly permeable bedrock is likely to allow a relatively large proportion of the precipitation received to percolate rapidly down to the water table. The remainder of the study area consists of alluvial valleys and alluvial or proluvial plains with thick and clayey soils, which cover chalky limestones and marls of the Paleocene/Eocene formation. Groundwater recharge in these regions is thought to be negligible, because the water holding capacity of the soil is high and the permeability of the

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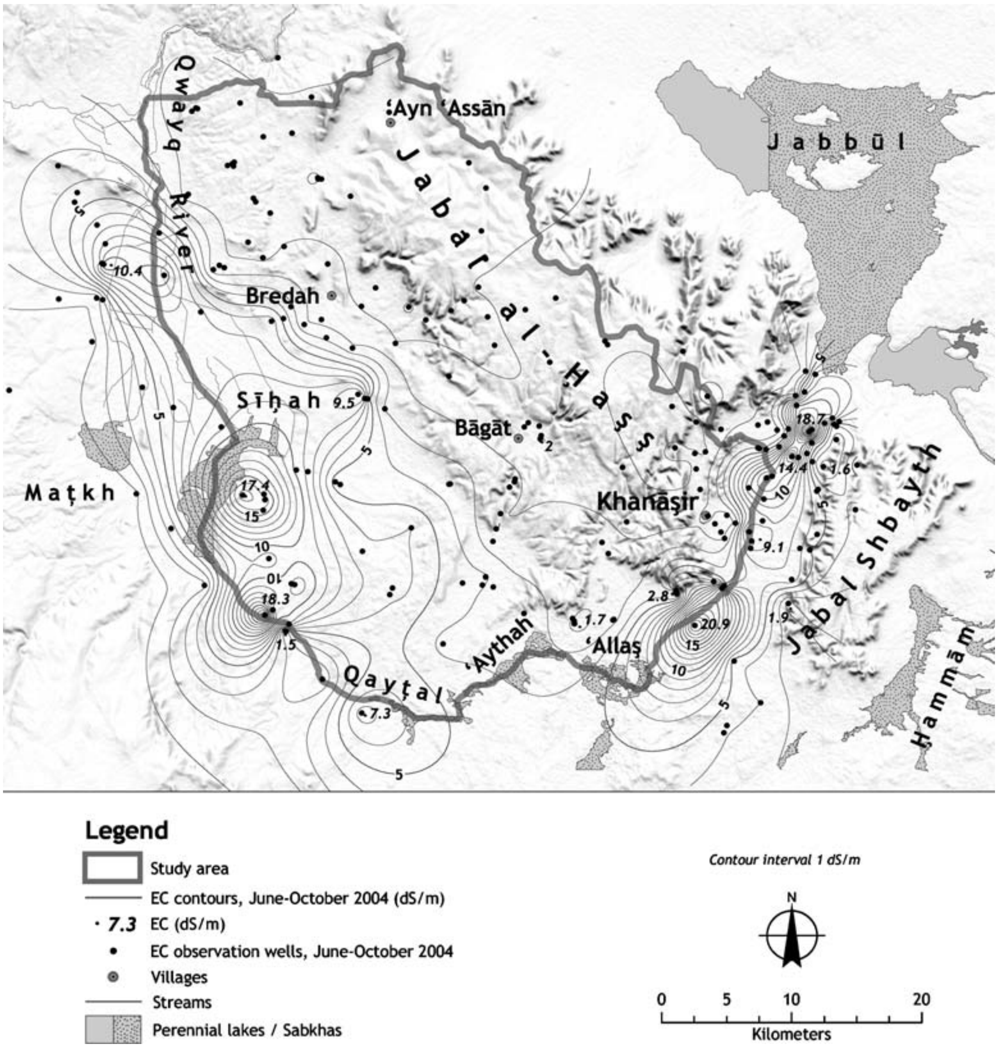

Fig. 6 Electrical conductivity of groundwater in the upper aquifer and the location of EC observation wells in the study area. *Contour lines* are given in dS m^{-1}

Table 2 Electrical conductivity (EC), pH, concentration of major ions and water type of precipitation samples in Aleppo and the south-eastern part of the study area

Date	EC dS m^{-1}	pH	K^+ meq L^{-1}	Na^+	Ca^{2+}	Mg^{2+}	SO_4^{2-}	HCO_3^-	NO_3^-	Cl^-	Water type
Aleppo[a]											
Dec-89	0.11	8.52	0.00	0.04	1.40	0.16	0.17	1.20	0.15	0.11	Ca–HCO_3
Jan-90	0.10	9.46	0.01	0.02	1.00	0.16	0.23	0.61	0.11	0.06	Ca–HCO_3
Umm Miyal[b]											
Jan-02				0.31[c]	1.15	0.22	0.46	0.85	0.10	0.25	Ca–HCO_3
Feb-02				0.10[c]	1.04	0.07	0.34	0.60	0.34	0.08	Ca–HCO_3
Feb-03	0.14	7.75		0.18[c]	0.92	0.07	0.14	0.70	0.44	0.12	Ca–HCO_3
March-03	0.13	7.88		0.17[c]	1.14	0.07	0.12	0.85	0.12	0.06	Ca–HCO_3
Banawi[b]											
Dec-02	0.084	7.55		0.09[c]	0.53	0.07	0.13	0.48	0.07	0.07	Ca–HCO_3
Feb-03	0.075	7.45		0.08[c]	0.57	0.03	0.15	0.46	0.08	0.06	Ca–HCO_3
March-03	0.08	7.46		0.06[c]	0.70	0.04	0.10	0.49	0.09	0.06	Ca–HCO_3

[a] According to Kattan (1996)

[b] According to Abou Zakhem (2008)

[c] $Na^+ + K^+$

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Table 3 EC, pH, concentration of major ions and water type of the groundwater samples collected from the study area

Well identifier (see Fig. 4)	EC	pH	K^+	Na^+	Ca^{2+}	Mg^{2+}	SO_4^{2-}	HCO_3^-	NO_3^-	Cl^-	Water type
	dS m^{-1}	meq L^{-1}									
Quaternary, lacustrine sediments											
288	10.59	7.54	0.2	64.7	13.6	27.5	33.0	4.3	1.4	73.9	Na–Cl
289	11.87	7.30	0.1	88.5	17.5	31.4	40.3	3.0	1.2	91.0	Na–Cl
321	8.83	7.79	0.7	39.7	26.7	32.7	52.7	2.7	1.2	36.2	Na–SO_4
358	15.74	7.80	0.3	81.4	43.9	52.7	66.5	2.3	0.4	99.2	Na–Cl
360	20.9	7.22	0.3	131.4	44.5	79.6	84.3	3.3	0.3	144.4	Na–Cl
Quaternary, proluvial sediments											
93	1.23	7.30	0.0	5.5	3.1	2.6	4.2	3.4	1.0	3.6	Na–SO_4/HCO_3/Cl
107	1.23	7.51	0.0	5.7	3.0	2.4	2.3	4.9	0.6	4.0	Na–HCO_3
143	2.87	7.32	0.1	13.0	9.7	7.0	12.0	3.2	0.4	12.4	Na–Cl/SO_4
146	2.64	7.18	0.1	13.0	9.3	7.4	16.8	4.1	0.5	9.1	Na–SO_4
152	3.02	7.56	0.1	10.8	11.6	9.3	12.2	1.3	2.0	15.6	Ca/Na–Cl
303	0.58	8.14	0.0	1.8	2.4	2.5	0.8	2.8	0.6	2.5	Mg/Ca–HCO_3
316	4.03	8.00	0.1	20.1	14.0	12.7	23.7	2.0	1.8	17.5	Na–SO_4
338	2.01	7.40	0.1	10.8	5.3	5.5	7.5	3.4	1.0	7.9	Na–Cl/SO_4
357	9.29	7.55	0.2	41.7	29.9	32.9	48.2	1.1	0.7	59.8	Na–Cl
Upper Miocene, basalts											
25	0.96	8.25	0.0	4.5	3.3	3.0	3.8	2.5	1.2	3.0	Na–SO_4
39	1.26	7.94	0.0	7.7	3.5	2.7	6.0	2.8	1.2	3.2	Na–SO_4
128	0.49	8.16	0.0	2.5	1.5	1.3	0.6	2.0	0.9	1.8	Na–HCO_3
129	0.5	8.12	0.0	1.8	1.8	1.5	0.8	2.1	0.9	1.8	Ca/Na–HCO_3
131	1.06	7.99	0.0	6.8	2.3	2.7	4.3	3.0	1.0	2.7	Na–SO_4
133	1.3	7.78	0.0	7.2	3.5	3.7	6.7	3.0	1.3	3.0	Na–SO_4
135	1.06	7.93	0.0	5.3	3.3	3.0	3.6	2.8	1.9	3.1	Na–SO_4/Cl/HCO_3
307	1.53	8.22	0.0	8.9	2.8	3.8	4.2	2.5	3.1	5.0	Na–Cl/SO_4
308	0.72	8.20	0.0	3.5	2.2	2.1	1.6	2.1	1.2	2.3	Na–HCO_3
309	1.41	8.12	0.1	7.1	4.2	3.3	6.3	2.3	1.3	4.8	Na–SO_4
312	1.27	8.20	0.0	5.5	4.9	3.3	5.1	1.5	2.2	4.0	Na/Ca–SO_4
KV.241[a]	0.4	7.83		0.9[b]	1.9	1.0	0.9	1.9	0.6	0.8	Na/Ca–Cl
Middle Miocene, Helvetian, limestones and basalts											
117	1.71	8.25	0.1	6.8	8.5	3.7	8.3	2.7	1.8	4.7	Ca/Na–SO_4
258	0.75	7.91	0.0	2.7	3.0	2.7	1.2	2.7	1.4	3.2	Mg–HCO_3/Cl
274	0.95	8.04	0.0	3.5	3.4	3.6	0.7	4.2	1.0	4.8	Mg/Na/Ca–HCO_3/Cl
283	1.07	8.03	0.0	3.0	3.8	4.4	1.8	3.3	1.8	4.7	Mg/Ca–Cl/HCO_3
304	0.54	8.10	0.1	0.3	3.9	2.4	0.3	1.9	1.7	2.3	Ca–HCO_3/Cl
335	0.54	7.91	0.0	1.8	2.4	2.1	0.8	2.4	0.5	2.0	Ca/Mg–Cl
354	6.49	7.82	0.1	39.7	19.5	16.6	24.1	2.8	1.3	46.9	Na–Cl
Paleogene, chalky limestones											
41	1.96	7.30	0.0	7.9	6.8	3.9	7.4	4.3	1.1	6.9	Na–SO_4/Cl
46	0.88	7.72	0.0	3.0	3.2	2.6	4.2	2.3	0.9	2.2	Ca/Na–SO_4
336	1.9	7.40	0.0	10.3	5.9	5.5	10.2	2.4	1.5	7.6	Na–SO_4
166	3.86	7.98	0.1	19.4	15.9	10.4	22.3	1.8	2.6	16.0	Na–SO_4
189	6.54	7.90	0.1	27.0	38.1	13.9	38.7	1.5	3.5	28.9	Ca–SO_4
KV.71[a]	0.65	7.78		3.0[b]	2.4	1.6	2.6	2.4	0.8	1.6	Na/Ca–Cl
KV.254[a]	2.33	7.38		9.3[b]	8.7	7.2	10.6	4.3	2.0	10.0	Na/Ca–Cl

[a] According to Abou Zakhem (2008)

[b] Na^++K^+

bedrock is low. The low groundwater Cl^- concentration observed in the Jabal Al Hass plateau and Helvetian outcrop areas (Table 3) suggests that the groundwater in this region has been much less affected by evaporation or transpiration than the groundwater in the other areas studied, which support the assumptions made about the location of recharge areas. In addition, high groundwater levels in the recharge regions, especially in the Jabal Al Hass plateau, indicate a flow of groundwater from these regions towards the valleys (Fig. 5). Recharge is the only mechanism that can explain a hydraulic gradient of this magnitude.

Chloride mass-balance calculations The parameter values used in the CMB method (Eq. 2) and their relative uncertainty are presented in Table 5. The calculated groundwater recharge rates (R*) in the Helvetian region (11 mm $year^{-1}$) and in the Jabal Al Hass (12 mm $year^{-1}$) were nearly equal. The long-term average groundwater recharge in the study area (R) was estimated to be 8.9×10^6 m^3 $year^{-1}$.

Net groundwater pumping

The relative uncertainties and parameter values used in the calculation of net groundwater pumping are presented in Table 6. Average farmer irrigation rates and the estimated aerial coverage of the different irrigated crops in the study area are given in Table 7. Multiplying these numbers by their areas showed that the annual amount of

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Table 4 Best, minimum and maximum estimates of the normalized difference vegetation index (NDVI) thresholds and aerial coverage of the corresponding irrigated fields in areas irrigated with groundwater (area A in Figs. 7 and 8), and irrigated fields in areas where both groundwater and surface water were used for irrigation (areas B and C in Figs. 7 and 8)

Satellite image	NDVI threshold estimates			Irrigated area estimates		
	'Best'	Min.	Max.	'Best' km^2	Min.	Max.
Winter, area A						
Landsat 5 TM, 01-April-1987	0.45	0.35	0.55	30.52	14.08	66.68
Landsat 5 TM, 07-April-1995	0.40	0.32	0.51	40.84	19.25	70.63
Landsat 7 ETM+, 21-March-2003	0.65	0.55	0.72	85.41	55.04	143.90
Winter, area B and C						
Landsat 7 ETM+, 21-March-2003	0.49	0.42	0.71	95.28	51.19	107.35
Summer, area A						
Landsat 2 MSS, 9-Sept-1975	0.43	0.42	0.47	15.36	11.98	16.39
Landsat 5 MSS, 23-July-1985	0.48	0.44	0.49	6.17	5.62	9.20
Landsat 7 ETM+, 24-Sept-1999	0.52	0.49	0.56	20.22	15.14	25.84
Landsat 7 ETM+, 10-Sept-2000	0.38	0.36	0.44	23.30	16.05	27.24
Summer, area B and C						
Landsat 2 MSS, 9-Sept-1975	0.47	0.42	0.56	17.43	11.32	25.42
Landsat 5 MSS, 23-July-1985	0.50	0.46	0.63	20.57	5.69	33.94
Landsat 7 ETM+, 24-Sept-1999	0.59	0.51	0.71	27.20	13.47	44.61
Landsat 7 ETM+, 10-Sept-2000	0.47	0.37	0.61	22.26	10.06	39.19

groundwater pumped for irrigated agriculture was 56.7×10^6 m^3 year^{-1}, which greatly exceeds the estimated recharge (Table 5). With 6.1×10^6 and 0.8×10^6 m^3 year^{-1} the volume of groundwater pumped for domestic and livestock use, respectively, was significantly smaller than the volume pumped for irrigation. The irrigation return flow, computed as a percentage of the total irrigation volume (including the contributions from the deep aquifer and surface water resources), was estimated as 6.4×10^6 m^{3}year^{-1}.

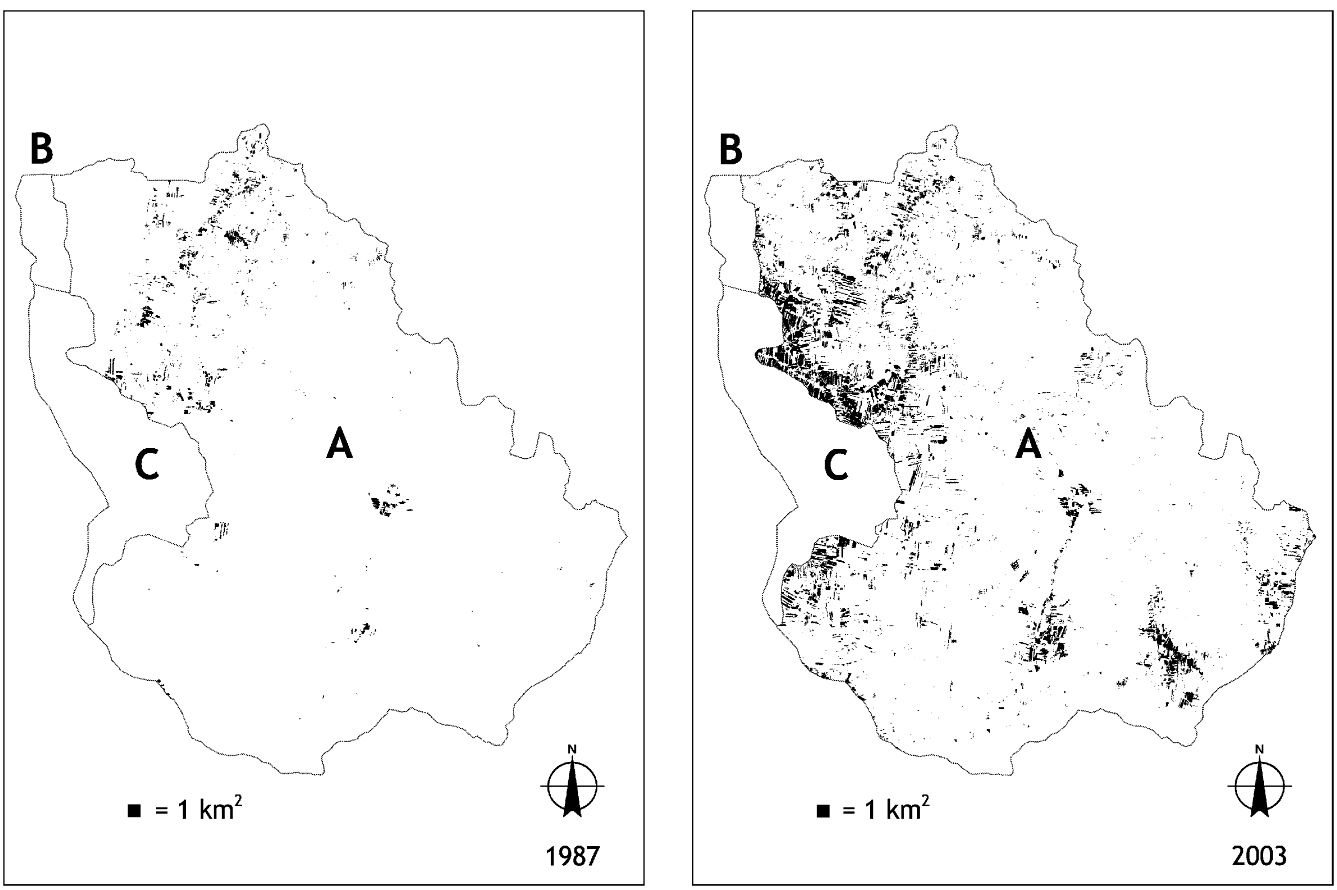

Fig. 7 Location of groundwater-irrigated fields within the study area in April 1987 and March 2003, based on the satellite image analysis. *A* is the area where only groundwater is used for irrigation, *B* and *C* represent areas that are irrigated with surface water in the winter season and both surface and groundwater in the summer season

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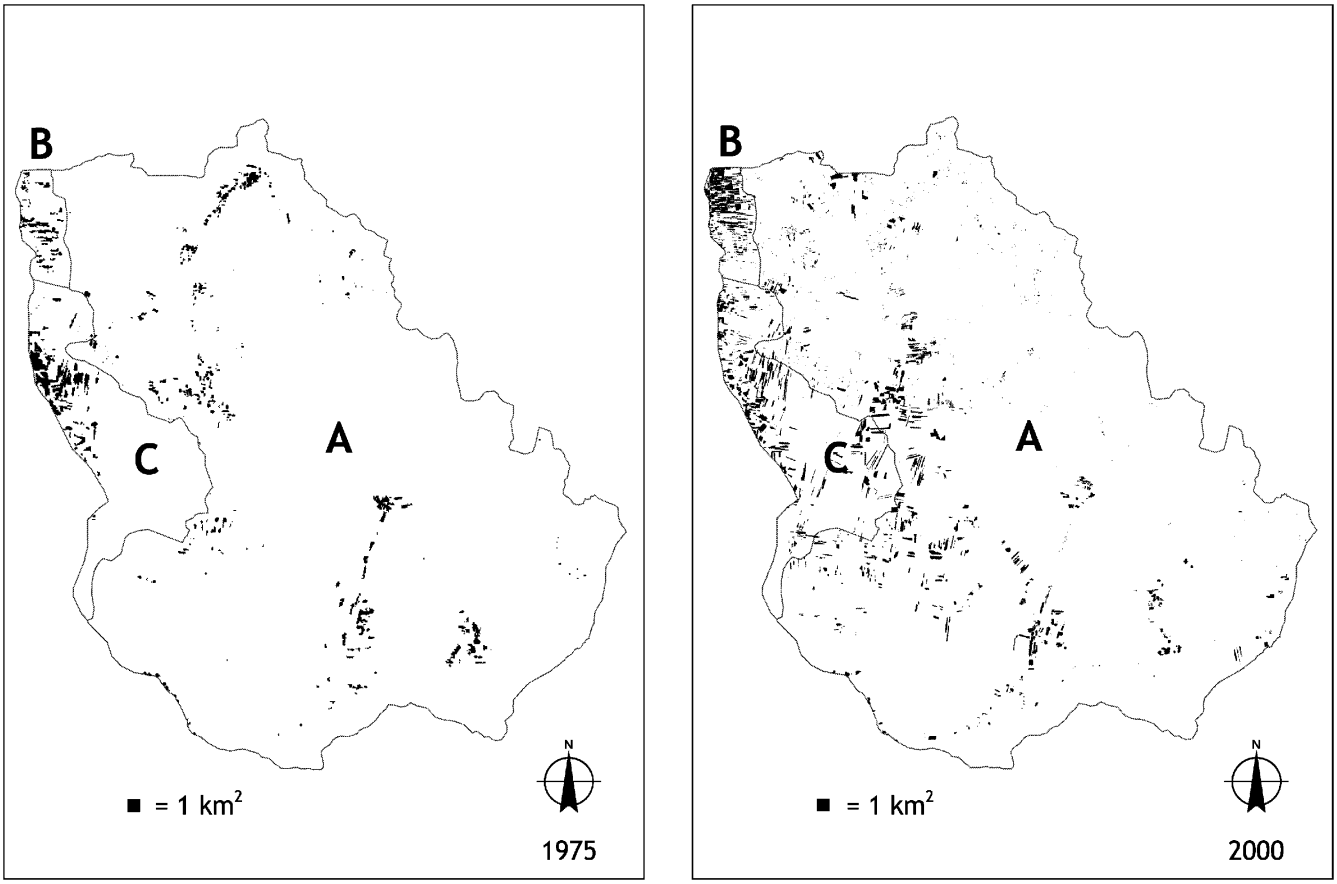

Fig. 8 Location of irrigated fields within the study area in September 1975 and September 2000, based on the satellite image analysis. *A* is the area where only groundwater is used for irrigation, *B* and *C* represent areas that are irrigated with surface water in the winter season and both surface and groundwater in the summer season

Final groundwater balance

The best estimates of the water balance components of the upper aquifer, as presented in Table 8, indicated an annual decrease in storage of 53×10^6 m^3. The maximum estimate of the change in storage was more than double the best estimate. Even the minimum estimate indicated an annual loss of 9.5×10^6 m^3 from the upper aquifer.

Table 5 Parameters used in the calculation of groundwater recharge, their best estimate, uncertainty and minimum and maximum estimates

Recharge region	Parameter	Best estimate	Estimated relative uncertainty	Minimum estimate	Maximum estimate
Helvetian limestone outcrop	Average Cl^- concentration of groundwater samples (mg L^{-1})	122	25%	92	153
	Average Cl^- concentration of precipitation samples (mg L^{-1})	3.4	See text	2.0	6.0
	Average precipitation (mm year^{-1})	302	10%	272	332
	Dry Cl^- deposition, summer (mg m^{-2})[a]	347	See text	69	693
	Recharge (mm year^{-1})	11		6.7	18
	Area of recharge (km^2)	232	10%	208	255
	Total recharge (10^6 m^3 year^{-1})	2.6		1.4	4.5
Miocene basalt outcrop (Jabal Al Hass plateau)	Average Cl^- concentration of groundwater samples (mgL^{-1})	105	25%	79	131
	Average Cl^- concentration of precipitation samples (mg L^{-1})	3.4	See text	2.0	6.0
	Average precipitation (mm year^{-1})	277	10%	249	305
	Dry Cl^- deposition, summer (mg m^{-2})[a]	347	See text	69	693
	Recharge (mm year^{-1})	12		7.2	19
	Area of recharge (km^2)	527	10%	474	580
	Recharge (10^6 m^3 year^{-1})	6.3		3.4	11
Total	Recharge (10^6 m^3 year^{-1})	8.9		4.8	16

[a] Total deposition during the 6 month dry season

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Table 6 Minimum, maximum and best estimates of the parameters used to calculate net pumping from the upper aquifer and their estimated uncertainty

Parameter	Best estimate	Estimated relative uncertainty	Minimum estimate	Maximum estimate
Irrigation				
Area irrigated with groundwater from the upper aquifer, summer (km^2)	26.23	See text	15.91	37.14
Area irrigated with groundwater from the upper aquifer, winter (km^2)	59.79	See text	38.53	100.73
Average water use, summer crops (mm)	1,230	25%	923	1,537
Average water use, winter crops (mm)	400	25%	300	500
Groundwater pumped for irrigated agriculture (10^6 m^3 $year^{-1}$)	56.2		26.2	107.5
Irrigation return flow				
Total irrigated area, summer (km^2)	45.56		26.11	66.43
Total irrigated area, winter (km^2)	180.69		106.23	251.25
Average irrigation application, summer crops (mm)	1,230	25%	923	1,537
Average irrigation application, winter crops (mm)	400	25%	300	500
Irrigation return flow (% of irrigation application)	5		10	0
Irrigation return flow (10^6 m^3 $year^{-1}$)	−6.4		−5.6	0
Domestic water supply				
Population	333,000	5%	316,350	349,650
Average domestic water use (L day^{-1})	50	25%	37.5	62.5
Groundwater pumped for domestic water use (10^6 m^3 $year^{-1}$)	6.1		4.3	8.0
Livestock				
Number of livestock	596,000	10%	536,400	655,600
Average water use (L day^{-1})	3.9	25%	2.9	4.9
Groundwater pumped for livestock (10^6 m^3 $year^{-1}$)	0.8		0.6	1.2
Total				
Net groundwater pumping (10^6 m^3 $year^{-1}$)	56.7		25.5	116.7

Discussion

Groundwater level changes

In addition to the decline of groundwater levels suggested by the comparison of historic groundwater level data (Table 1), the cone of depression in the Qwayq River Valley indicated that groundwater levels have declined considerably in this location as well. The development of the depression cone is likely to have commenced in the 1970s. In the 1960s, before motorized pumps were introduced to the area, the groundwater in the Qwayq River Valley was located close to the surface and drained towards the Qwayq River (Wolfart 1966). By the early 1980s, however, groundwater levels had declined to a level below the base of the Qwayq River. According to Gruzgiprovodkhoz (1982), the groundwater level at a location 1-km east of the Qwayq River was approximately 240–250 masl, which is between 5 to 15 m below the surface of the river. At ICARDA's experimental farm, located 6 km west of the Qwayq River, the groundwater level was 35 m below the water level of the river in 1983 (ICARDA, unpublished data, 2006).

In general, the drawdown of groundwater levels since the 1960s seems to be greatest in the more productive aquifers, e.g., the relatively permeable Helvetian formation and the fractured Paleocene/Eocene limestone in Bagat Valley, in spite of the greater storativity of these aquifers. These aquifers seem to have fallen victim to the fact that productive aquifers (with high transmissivity and storativity) permit high pumping rates and encourage intensive exploitation and rapid depletion. Low-yielding aquifers, on the other hand, limit groundwater abstraction to rates that are more likely to be sustainable in the long term.

Table 7 Crops, aerial coverage, and estimated average irrigation water use

Crop type	Aerial coverage %	Average irrigation application mm $year^{-1}$
Summer crops		
Cotton	52	1,500
Maize	28	850
Sunflower	5	570
Sesame	3	560
Sugar beet	2	920
Vegetables[a] and melon	10	1,500
Weighted average		1,230
Winter crops		
Wheat	99.9	400

[a] Tomato, pepper, eggplant, cucumber

Hydrochemistry

The high salinity of the groundwater in the Qwayq Valley and the depressions is probably the result of evapotranspiration of groundwater, as suggested earlier by Wolfart (1966). No halite was reported in the sediments in the Qwayq Valley (Gruzgiprovodkhoz 1982); therefore, all of the high concentrations of Cl^- found there must have resulted from evapotranspiration of groundwater located at shallow depths. This fact supports the observation by Wolfart (1966) that the groundwater level in the Qwayq Valley was formerly located close to the surface.

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Table 8 Best, minimum and maximum estimates of the groundwater balance components, resulting in best, minimum and maximum estimates of change in storage

Water balance component	Best estimate	Minimum estimate of storage change	Maximum estimate of storage change
	10^6 m^3 year^{-1}		
Groundwater recharge (R)	8.9	16	4.8
Net pumping (NP)	−56.7	−25.5	−116.7
Evaporation (ET)	−4.5	0	−4.8
Cross-boundary outflow (Q)	−0.2	−0.0	−1.5
Change in storage (*dV*/*dt*)	−53	−9.5	−118

The SO_4/Cl ratios in the Paleogene and Quaternary geological units are related to their respective recharge areas, which is tentative evidence that both the Helvetian outcrop region and the Jabal Al Hass region are important sources of recharge for the groundwater in the adjacent valleys. Those wells in the Qwayq Valley that are close to the Miocene Helvetian outcrop (wells 107, 288, 289 and 303 in Fig. 4) exhibit an average SO_4/Cl ratio of 0.4 (0.3–0.6), while the average SO4/Cl ratios found in the Helvetian outcrop region is 0.5. The areas close to the Jabal Al Hass plateau show higher SO_4/Cl ratios (0.6–1.9), which fall within the range of values on the Jabal Al Hass plateau itself (0.3–2.2). In some of these wells (321, 358 and 360), the SO_4/Cl ratio could have been affected by dissolution of gypsum or gypseous loams contained in the lacustrine Quaternary sediments. However, the SO_4/Cl ratios in these samples (0.6–1.5) fall within the range of the ratios of samples taken in the upstream areas (0.3–2.2), which suggests that the influence of gypsum dissolution on groundwater composition is low. With an average SO_4/Cl ratio of 1.2, the SO_4/Cl ratios observed in the Jabal Al Hass area are significantly higher than on the Helvetian outcrop region, where the average ratio is 0.5. This might be explained by the additional dissolution of iron-sulfide minerals (pyrite) that are contained in the Paleocene/Eocene limestones prevalent in the Jabal Al Hass plateau. The fact that the SO_4/Cl ratios in the northern part of the Jabal Al Hass area (wells 128 and 129) are similar to those found in the Helvetian outcrop region probably results from the presence of the Helvetian formation under the basalt sheet in the northern part of the Jabal Al Hass area.

The spatial distribution of the Na/Cl ratios showed a different picture than that of the SO_4/Cl ratios. Because the flowlines of the wells in the southern part of the valley originate in the Jabal Al Hass region, one would expect Na/Cl ratios to be in the same range as those found in the Jabal Al Hass region. However, with the exception of well 93, the wells in the Qwayq Valley exhibit Na/Cl ratios of 0.7–1.1, while Na/Cl ratios in the Jabal Al Hass region range from 1.0 to 2.5. The wells located to the south of the Jabal Al Hass were found to have Na/Cl ratios that ranged from 0.9 to 1.6, which are similar to those of the wells located in the Jabal Al Hass area. The relatively high Na/Cl ratios in the Jabal Al Hass region are probably caused by the dissolution of Na from sodium rich feldspar minerals that are found in the upper Miocene basalt sheet (Technoexport 1966). The reason behind the lower Na/Cl ratios in the Qwayq Valley is unknown. A variety of processes could have influenced this ratio such as interaction between groundwater and water from the Qwayq River and cation exchange processes.

Development of irrigated agriculture

The results of this study indicated that the groundwater-irrigated area has approximately doubled from the mid 1980s to the present. The increase in the winter season was slightly larger than the summer season. The reason behind this is probably the fact that not all wells yield sufficient water for full irrigation of summer crops.

In 1975, groundwater irrigation was restricted to a few intensively irrigated areas near Ayn Assan, Bagat and Siyalah. However, use of irrigation at these locations actually declined over the years. In fact, the overall increase observed in total irrigation water use between 1975 and 2003 was caused by an increase in irrigation in areas outside those 'hotspots'. In general, well yields in these 'hotspots' are relatively high (3–13 L s^{-1}) while the groundwater in these areas has a low salinity (EC <3 dS m^{-1}). The high well yields observed in Bagat Valley and Siyalah Valley are probably the result of fracturing of the Paleocene/Eocene chalky limestone formation, as they contrast sharply with the generally low yields (1–2 L s^{-1}) obtained from wells in surrounding areas that tap the same formation. High well yields in the area of Ayn Assan are related to the relatively high permeability of the Helvetian limestone formation.

The decrease in irrigated area observed in the areas around the villages of Bagat and Siyalah during the period 1975–2003 most probably resulted from declining groundwater levels, which resulted in a fall in well capacity, an increase in pumping costs and dried up wells. Interviews with farmers in Bagat and Siyalah indicated that a number of farmers have now given up irrigated agriculture and have returned to the traditional rain-fed barley-livestock system, while a significant number of other people have coped by migrating to cities or to other countries. The general increase in groundwater-based irrigation outside these areas during the period 1975–2003 was driven by the availability of drilling equipment and the low (subsidized) price of diesel fuel used for running the pumps.

Groundwater balance components

Recharge

The CMB method requires that the Cl^- concentration of groundwater should only be determined by the Cl^-

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concentration of groundwater recharge. It can be assumed with a reasonable level of confidence that this was the case for the recharge areas (the Jabal Al Hass plateau and the Helvetian limestone outcrop region), because water derived from precipitation is unlikely to have mixed with saline water from the valleys because of the significant hydraulic gradient (approximately 10–25 m km^{-1}) present between the recharge areas and the surrounding valleys (Fig. 5). Furthermore, groundwater salinity levels in the recharge areas are relatively homogenous, and no salinity gradient between the recharge areas and the surrounding valleys could be observed (Fig. 6). It can also be assumed that the computed recharge reflects the 'natural' recharge and is not affected by irrigation return flow, because irrigated agriculture in the recharge areas is limited to a few isolated fields and home gardens (see Figs. 7 and 8).

The CMB method cannot be used in regions where groundwater salinity is influenced by paleo-hydrological conditions or evaporation of groundwater or surface water, as is often the case in depressions or sabkhas. Therefore, the method is not suitable for use in many areas in semi-arid regions, including those in the lower parts of the study area. However, in general, such areas are covered by lacustrine sediments with a low permeability, so the potential for recharge is rather limited. In addition, many such areas serve as natural discharge areas, and therefore groundwater recharge can be assumed to be negligible. Thus, using the CMB method to estimate recharge values in the more elevated areas can provide estimates of the total groundwater recharge in a semi-arid basin.

Final groundwater balance

The computed water balance indicated that under present pumping rates, groundwater resources are being depleted by a rate that lies between 9.5×10^6 m^3 $year^{-1}$ and 118×10^6 m^3 $year^{-1}$ (Table 8). The high uncertainty range in this figure is mainly related to the difficulty in estimating the irrigation water use. This suggests that improved monitoring of agricultural water use is critical for the reduction of this uncertainty, while it would at the same time facilitate measures to reduce agricultural water use, which could stabilize groundwater levels.

The computed depletion is larger than the estimated current rate of evapotranspiration, so even if declining groundwater levels would result in a decrease in evapotranspiration in the future, recharge and discharge would still not be in balance. The volume of surface water available to compensate for the computed imbalance between recharge and discharge from the aquifer is limited. Virtually all the water from the Qwayq River is used for irrigation. Inflows from adjacent parts of the basin are also unlikely to be able to compensate for the imbalance. The fact that there is an outflow of groundwater across the western boundary of the study area indicates that the groundwater system of the adjacent region is under greater stress than that of the study area, while to the south and east, the study area is bordered by areas that receive less than 200 mm rainfall per year.

As long as discharge exceeds recharge, the groundwater system will continue to be in a state of disequilibrium, which will result in further depletion of the groundwater resources. Declining groundwater levels will increase the cost of groundwater abstraction, as wells will have to be dug deeper and more fuel will be needed to pump water from greater depths. Moreover, well yields are likely to decline irrespective of the deepening of wells, because the productivity of the Paleocene/Eocene and Helvetian formations is determined largely by fractures, which often decrease in size with increasing depths, due to an increase in overburden pressure. In addition, the continued lowering of groundwater levels in the valleys could draw saline groundwater from the adjacent sabkhas and the Qwayq Valley into those parts of the aquifer that previously contained freshwater, thereby further endangering groundwater based irrigated agriculture, as well as domestic water supply.

Bredehoeft et al. (1982) and Bredehoeft (2002) argue that sustainable pumping rates are independent of recharge, and actually depend on capture (the sum of the change in recharge and the change in discharge caused by pumping). However, in most arid and semi-arid basins recharge is not affected by pumping, as the groundwater level in areas where recharge takes place is usually found deep below the ground surface. Furthermore, in these dry environments, induced seepage from surface water resources or a reversal of boundary conditions are not very likely to provide a substantial inflow to the system. In these cases, the concept of capture is only relevant as long as the amount of groundwater pumped does not exceed the amount of recharge. Thus, comparing the amount of groundwater pumped with the volume of any other groundwater outflows and recharge can provide information on (1) whether or not an aquifer is currently being mined, and (2) the magnitude of pumping rates that would be sustainable.

To stabilize the groundwater levels in the study area, the first step would be to reduce pumping rates to a value that equals the recharge. This value is here referred to as the maximum sustainable pumping rate. The estimated maximum sustainable pumping rate, which ranges from 4.8×10^6 to 16×10^6 m^3 $year^{-1}$, is likely to be sufficient to cover the current water needs for domestic and livestock water supply, which was estimated to lie between 4.9×10^6 and 9.2×10^6 m^3 $year^{-1}$ (Table 6). However, the volume of water that would be left for irrigation ranged between 0 and 11×10^6 m^3 $year^{-1}$, which is much lower than the estimates of water currently pumped for irrigation, 26×10^6–107×10^6 m^3 $year^{-1}$ (Table 6). Therefore, future stabilization of groundwater levels would require significant decreases in groundwater-irrigated agriculture. It should be noted that the estimated sustainable pumping rates do not take into account potential effects of the pumping on the outflows to the sabkhas or the groundwater quality. Avoiding adverse effects on these flows or the groundwater quality would most likely require lower pumping rates than the maximum sustainable pumping rate.

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Conclusions

A water-balance study can be a useful method for groundwater-resource assessments in semi-arid catchments with limited data and resources. Although sustainable groundwater development does not involve stabilizing groundwater levels alone, the latter is usually the first priority in groundwater reservoirs that are mined. The water-balance approach such as that applied in this study can yield information on whether a groundwater reservoir is being mined and on the magnitude of pumping rates that allow stabilization of the groundwater levels. The method can be applied in regions where (1) surface water resources are absent or limited and (2) recharge is independent of groundwater levels. These conditions apply especially to semi-arid catchments, where surface water resources are generally limited and groundwater levels are usually only found at depth.

As has been shown in many previous studies, the CMB method can be applied to provide estimates of regional, long-term average rates of groundwater recharge in semi-arid regions. This study demonstrated that the use of the method requires a thorough knowledge of the hydrogeology of the selected study area, because (1) the results obtained depend upon the selection of an appropriate conceptual model of recharge and the delineation of the recharge areas, based on all available hydrogeological information, and (2) a number of hydrogeological conditions have to be met for the application of the method to be valid.

Information obtained through the water-balance study was complemented by analyses of the growth in groundwater pumping and of past and present groundwater levels, which provided insights into the response of the groundwater system to pumping. Comparing present groundwater levels with past observations indicated that groundwater levels in the Jabal Al Hass region have declined significantly over the last 30 years. The average drop in the water levels in 13 observation wells was 23 m. In the Qwayq River Valley, the water table is no longer connected to the Qwayq River, as it has declined to a depth of 70–80 m below the surface over the last 30–40 years. This decline coincided with an increase since the mid 1980s in the groundwater-irrigated area from 1.7 to 2.9% of the study area in the summer season and from 1.9 to 5.3% in the winter season.

The water balance calculated for the study area showed that groundwater resources are being depleted by a rate that lies between 9.5×10^6 and 118×10^6 m^3 $year^{-1}$. The imbalance between discharge and recharge is too large to allow it to be compensated for by inflow from surface water resources, a reversal of boundary conditions, or a decrease in natural discharge. With 56.7×10^6 m^3 $year^{-1}$, net pumping was the largest groundwater balance component, but the large range between the minimum and maximum estimates (25.5×10^6 and 116.7×10^6 m^3 $year^{-1}$) indicated that this component is difficult to quantify. The estimated maximum sustainable pumping rate is likely to be sufficient to sustain current domestic and livestock use.

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